1.1 Introduction

From messages in bottles to exotic tropical seeds washing up on temperate shores (Guppy 1917; Muir 1937), the dispersal of floating debris at sea has long fascinated people. As early as 1870 Jules Verne provided a graphic description of how floating debris accumulates in ocean gyres in the chapter on the Sargasso Sea in his famous novel *Twenty Thousand Leagues under the Sea.* However, this review focuses on the last 50 years because from the perspective of environmental impacts the history of marine litter research is closely linked to the development of plastics. Plastics are a diverse group of synthetic polymers that have their origins in the late 19th century, but which really came to the fore in the mid-twentieth century. Their low density, durability, excellent barrier properties and relatively low cost make plastics ideal materials for a wide range of manufacturing and packaging applications. Their versatility has seen the amount of plastic produced annually increase rapidly over the last few decades to an estimated 288 million tonnes in 2012 (Fig. 1.1), and this total continues to grow at about 4 % per year (PlasticsEurope 2013). However, the properties that make plastics so useful also make inappropriately handled waste plastics a significant environmental threat. Their durability means that they persist in the environment for many years, and their low density means that they are readily dispersed by water and wind, sometimes travelling thousands of kilometres from source areas (Ryan et al. 2009). As a result, plastic wastes are now ubiquitous pollutants in even the most remote areas of the world (Barnes et al. 2009).

Over the last 60 years we have seen a major shift in perception surrounding the use of plastics, especially in one-off applications. Once seen as the savior of the American housewife (Life Magazine 1955), there are now calls to treat waste plastics as hazardous materials (Rochman et al. 2013a), reiterating a point first made by Bean (1987) that persistent plastic wastes qualify as hazardous wastes under the US Resource Conservation and Recovery Act. Most of the threats posed by plastics occur at sea (Gregory 2009; Thompson et al. 2009), where waste plastics tend to accumulate (Barnes et al. 2009; Ryan et al. 2009). This chapter briefly summarises the history of marine litter research. Trends in the numbers of

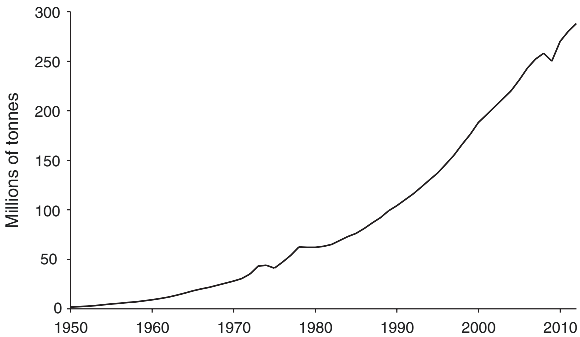

Fig. 1.1 Growth in global plastic production from 1950 to 2012 (millions of tonnes, adapted from PlasticsEurope 2013)

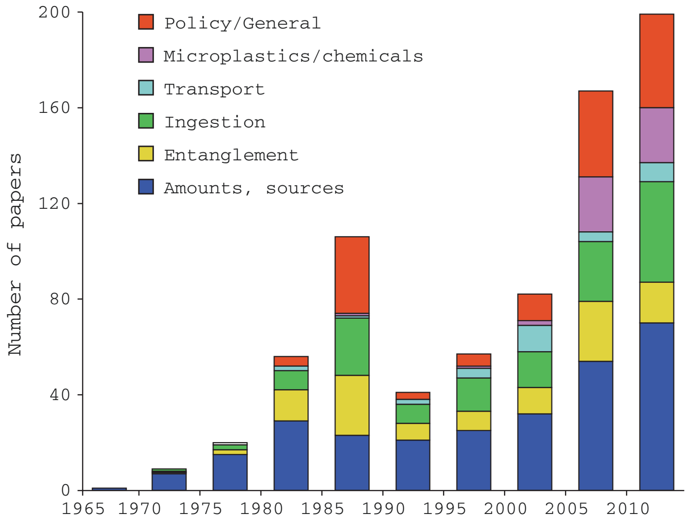

Fig. 1.2 Numbers of papers on different aspects of the marine litter issue published in five-year intervals over the last 50 years (based on a Web of Science search and unpublished bibliography; note that the final column only covers three years, 2011–2013)

papers on the marine litter problem (Fig. 1.2) show the growth in research from its infancy in the late 1960s, when it was still treated largely as a curiosity, through the 1970s and 1980s, when most of the threats to marine systems were identified, baseline data were collected on the distribution, abundance and impacts of marine litter, and policies were formulated to tackle the problem. Research tapered off in the 1990s, despite ongoing increases in the amounts of marine litter (Ryan and Moloney 1990, 1993), and it is only in the last decade or so that there has been a resurgence in research interest, following alarming reports of mid-ocean 'garbage patches' (Moore et al. 2001) and increasing appreciation of the pervasive nature of very small 'microplastic' particles (<0.5 mm) and their potential impacts on the health of marine ecosystems (Oehlmann et al. 2009; Thompson et al. 2009).

1.2 Seabirds and Seals—The First Signs of Trouble

Interactions between marine organisms and persistent litter were first recorded in the scientific literature in the late 1960s, when Kenyon and Kridler (1969) reported the ingestion of plastic items by Laysan Albatrosses (*Phoebastria immutabilis*) on the northwest Hawaiian Islands. They found plastic in the stomachs of 74 of 100 albatross chicks that died prior to fledging in 1966, with up to 8 items and an average of 2 g plastic per bird. However, this was an order of magnitude less than the average mass of pumice, seeds, charcoal and wood that the chicks also were fed by their parents. Kenyon and Kridler (1969) inferred that these indigestible items were swallowed inadvertently at sea, because virtually all items floated in seawater. They also speculated that the large size of many of the items might have contributed to the chicks' deaths by blocking their digestive tracts.

In fact, there were earlier records of seabirds ingesting plastics, with plastic found in stranded prions (*Pachyptila* spp.) in New Zealand as early as 1960 (Harper and Fowler 1987), and in Leach's storm petrels (*Oceanodroma leucorhoa*) from Newfoundland, Canada, in 1962 (Rothstein 1973). Non-breeding Atlantic puffins (*Fratercula arctica*) collected from 1969 to 1971 were reported to contain elastic threads in their stomachs (Berland 1971; Parslow and Jefferies 1972). In some birds, these threads had formed tight balls up to 10 mm across, filling the gizzard and possibly partially blocking the pyloric valve leading into the intestine (Parslow and Jefferies 1972). Parslow and Jefferies (1972) noted that ingesting rubber and elastic was common among scavenging birds such as gulls, but that they regularly regurgitated such items along with other indigestible prey remains, implying that this was not a problem for such birds. And it was not just seabirds at risk. By the late 1950s there were records of marine turtles ingesting plastic bags, sometimes resulting in their deaths (Cornelius 1975; Balazs 1985). A mass of fishing line and other fishing gear blocked the intestine of a manatee (*Trichechus manatus*) in 1974 (Forrester et al. 1975), and stranded cetaceans were found to have eaten plastic by the mid-1970s (Cawthorn 1985).

Records of entanglement of marine organisms in plastic litter also started to increase in the 1960s. There were reports of birds and seals entangled in man-made items before this (e.g. Jacobson 1947), but they tended to remain in the gray literature (Fowler 1985; Wallace 1985). By 1964 northern fur seals (*Callorhinus ursinus*) were often reported entangled in netting and other artefacts in the Bering Sea, and the incidence of entangled seals harvested in the Pribilof Islands showed a steady increase from less than 0.2 % of the population in 1967 to a peak of over 0.7 % in 1975 (Fowler 1987). The entanglement rate then stabilized at around 0.4 % through the late 1970s and early 1980s (Fig. 1.3), but this was still sufficient to help to drive a population decrease in this species (Fowler 1987; Fowler et al. 1990). By comparison, entanglement rates of three seal species at the Farallon Islands off central California showed a marked increase in the early 1980s (Hanni and Pyle 2000).

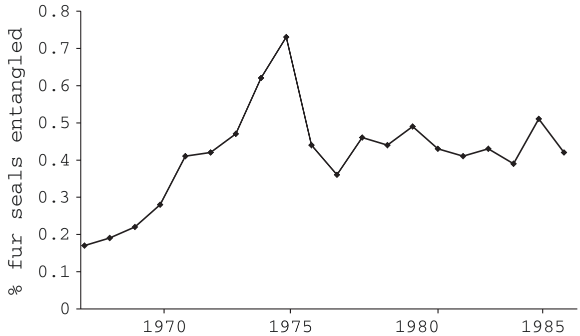

Fig. 1.3 Trends in the percentage of northern fur seals entangled on St. Paul Island, Alaska (adapted from Fowler et al. 1990)

Entanglement of fish and dogfish in rubber bands was reported in 1971 (Anon 1971; Berland 1971), and Gochfeld (1973) highlighted the entanglement threat posed by marine litter to coastal birds. Based on observations on Long Island in 1970 and 1971, Gochfeld (1973) reported how adult and chick black skimmers (*Rhynchops niger*) and two species of terns died after being entangled in nylon fishing line, kite strings, six-pack holders, bags and bottles. Although the numbers of birds affected were not great, Gochfeld (1973) argued that they might be sufficient to cause at least some populations to decrease, especially when combined with other human impacts in the region. Subsequently, Bourne (1976, 1977) summarised what was known about the threat posed by plastic ingestion and entanglement to seabirds, and reported how the incorporation of rope and netting in seabird nests can entangle and kill seabird chicks. He also highlighted the threat posed by the switch to manufacturing nets and other fishing gear from persistent polymers, including ghost fishing by lost or discarded gear (Bourne 1977). Entanglement was a significant cause of mortality for northern gannets (*Morus bassanus*), affecting roughly a quarter of birds found dead in the North Sea in the 1980s (Schrey and Vauk 1987), and remains a problem for this species today (Rodríguez et al. 2013).

1.3 The Early 1970s—Pellets and Other Problems in the North Atlantic

Many of these early records of ingestion and entanglement only came to light after two seminal papers on the occurrence of plastic particles at sea in the northwest Atlantic Ocean appeared in the leading journal *Science* in 1972. In the first paper, Carpenter and Smith (1972) reported the presence of plastic pellets and fragments in all 11 surface net samples collected in the western Sargasso Sea in late 1971, at an average density of around 3500 particles km^{-2} (290 g km^{-2}). Interestingly, the density of plastic was lowest towards the edge of the Sargasso Sea, where it bordered the Gulf Stream, suggesting that these particles had been accumulating in the North Atlantic gyre for some time (cf. Law et al. 2010; Lebreton et al. 2012; Maximenko et al. 2012). Carpenter and Smith (1972) noted that the plastic particles provided attachment sites for epibionts, including hydroids and diatoms, and speculated that such particles could become a significant problem if plastic production continued to increase. They also suggested that plastic particles could be a source of toxic compounds such as plasticisers and polychlorinated biphenyls (PCBs) into marine food webs.

In the second paper, Carpenter et al. (1972) reported high densities of polystyrene pellets in coastal waters off southern New England, east of Long Island (average 0.0–2.6 pellets m^{-3}, exceptionally reaching 14 pellets m^{-3}). Polystyrene is denser than seawater, so the pellets were not expected to disperse far from source areas, but some contained air-filled vacuoles, allowing them to float. The pellets supported communities of bacteria, and were found to have absorbed polychlorinated biphenyls (PCBs) from seawater. Pellets were recorded in the stomachs of eight of 14 fish species and one chaetognath (*Sagitta elegans*) sampled in the area. The fish ignored translucent pellets, only eating opaque white pellets, which suggested selective feeding on the more visible pellets. With up to 33 % of individuals of some fish species affected, Carpenter et al. (1972) raised concerns about the possible impacts due to intestinal blockage of smaller individuals as well as pellets being a source of PCBs.

In fact, Carpenter's two *Science* papers were not the first papers to describe small pieces of plastic litter at sea. Buchanan (1971) reported densities of up to 10^5 synthetic fibres m^{-3} in water samples from the North Sea, and larger fragments were reported to occur in "embarrassing proportions" in plankton samples. And although Heyerdahl (1971) mainly concentrated on oil and tar pollution, he reported sightings of plastic containers throughout the second *Ra* expedition across the North Atlantic. However, Carpenter's papers focused scientific attention on the ubiquitous nature of small plastic particle pollution at sea, and identified three possible impacts: intestinal blockage and a source of toxic compounds from ingested plastic, and the transport of epibionts.

Following Carpenter et al. (1972), large numbers of polystyrene pellets were reported from coastal waters in the United Kingdom (Kartar et al. 1973, 1976; Morris and Hamilton 1974) where they were ingested by three species of fish and a marine snailfish (*Liparis liparis*). More than 20 % of juvenile flounder (*Platichthys flesus*) contained ingested plastics, with up to 30 pellets in some individuals. Hays and Cormons (1974) found polystyrene pellets in gull and tern regurgitations collected on Long Island, New York, in 1971. Although the gulls may have consumed the plastic pellets directly while scavenging, their presence in the diet of terns almost certainly indicated that they were consumed in contaminated fish prey, providing the first evidence of trophic transfers of small plastic items. Sampling close to wastewater outfalls confirmed that the pellets came from plastic manufacturing plants (Hays and Cormons 1974). Fortunately, these point sources were fairly easy to identify and address. By 1975 the incidence of plastic ingestion by fish and snails in the UK's Severn Estuary had fallen to zero, indicating that the release of polystyrene pellets had virtually ceased from the manufacturing plants (Kartar et al. 1976). However, spillage of pellets by converters and during transport proved more difficult to contain.

Carpenter's two *Science* papers in 1972 stimulated a broader interest in marine litter and its impacts. Colton et al. (1974) reported a much more extensive survey of floating plastics in the North Atlantic and Caribbean. They showed that both industrial pellets and fragments of manufactured items occurred throughout the region, but were concentrated close to major land-based sources along the US eastern seaboard. Unlike Carpenter et al. (1972), they failed to find any plastics in fish sampled. Feeding trials with polystyrene pellets showed that juvenile fish seldom ingested plastics, and those pellets that were ingested seemingly passed through the fish with little impact.

Beach litter also came under increased scrutiny. Scott (1972) debunked the notion that beach users were responsible for most litter. He examined the litter found on inaccessible Scottish beaches that have few if any visitors, and inferred that most litter came from shipping and fisheries operating in the area. Initial studies of beach litter simply assessed standing stocks (Ryan et al. 2009); Cundell (1973) was the first researcher to report the rate of plastic accumulation. Working on a beach in Narragansett Bay, USA, he assessed the amount of litter washing ashore over one month. The first study of beach litter dynamics was conducted in Kent, United Kingdom, from 1973 to 1976. Dixon and Cooke (1977) showed that the weekly retention rate of marked bottles and other containers varied depending on the type of beach, and that plastic bottles remained on beaches longer than glass bottles. Strong tidal currents resulted in low retention rates (11–29 % per week) and transported litter throughout the southern North Sea. Some marked bottles travelled >100 km in one week, and others reached Germany and Denmark within 3–6 weeks. Dixon and Cooke (1977) also used manufacturer's codes to assess the longevity of containers and found that few (<20 %) were manufactured more than two years prior to stranding.

In addition to the growing awareness of plastic litter at the sea surface and stranded on beaches, the mid-1970s also saw the first records of plastics on the seabed. Holmström (1975) reported how Swedish fishermen "almost invariably" caught plastic sheets in their trawl nets when fishing in the Skagerrak. Subsequent analysis showed this to be low-density polyethylene, similar to that used for packaging. The samples, obtained from the seabed 180–400 m deep, were encrusted with a calcareous bryozoan and a brown alga (*Lithoderma* sp.). Holmström (1975) surmised that these encrusting biota had increased the density of the plastic sheets sufficiently to cause them to sink to the seabed. The bryozoan and brown alga typically occur in water <25 m deep, and the size of bryozoan colonies suggested that the plastic sheets had spent 3–4 months drifting in the euphotic zone close to the sea surface before sinking to the seabed. Subsequent trials confirmed that most plastics sink due to fouling (Ye and Andrady 1991), and trawl surveys and direct observations have confirmed that plastics and other persistent artefacts now occur on the seabed throughout the world's oceans (Barnes et al. 2009). Indeed, Goldberg (1994, 1997) suggested that the seabed is the ultimate sink for plastics in the environment, and plastic items typically comprise >70 % of seabed artefacts (Galgani et al. 2000). The Mediterranean Sea supports particularly high densities of litter on the seafloor, locally exceeding 100,000 items km^{-2}, and has been the subject of numerous studies to ascertain the factors determining the distribution and abundance of this litter (e.g. Bingel et al. 1987; Galil et al. 1995; Galgani et al. 1995, 1996). Interestingly, although benthic litter tends to concentrate around coastal cities and river mouths, the density of litter is often greater in deep waters along the continental shelf edge than in shallow, inshore waters due to the decrease in bottom currents offshore (Galgani et al. 1995, 2000; Barnes et al. 2009; Keller et al. 2010).

Winston (1982) elaborated on Carpenter and Smith's (1972) suggestion that plastic debris greatly increased settlement opportunities for organisms that live on objects floating at the sea surface. In particular, the bryozoan *Electra tenella* appeared to have extended its range and greatly increased in abundance in the western Atlantic Ocean. Subsequent research has highlighted the potential threat posed by drifting litter transporting organisms outside their native ranges (Barnes 2002; Barnes and Milner 2005; Gregory 2009). This is a serious problem, especially in remote regions, and can result in the transfer of potentially harmful organisms (Masó et al. 2003). However, it probably pales into insignificance in most regions compared to the transport by shipping and other human-mediated vectors (Bax et al. 2003), which in extreme cases can transfer entire communities across ocean basins (Wanless et al. 2010).

1.4 Shifting Focus to the North Pacific Ocean

Indications that the North Pacific was a hot spot for plastic litter date back to Kenyon and Kridler's (1969) paper on plastic ingestion by Laysan albatross. Subsequently, Bond (1971) found plastic pellets in all 20 red phalaropes (*Phalaropus fulicarius*) examined when many individuals of this species came ashore along the coasts of southern California and Mexico in 1969. The birds apparently starved due to a shortage of surface plankton, and some were observed feeding along the strand line where plastic pellets were abundant (Bond 1971). It was unclear whether this had contributed to the high incidence of plastic in these birds, but Connors and Smith (1982) found plastic in six of seven red phalaropes killed by colliding with powerlines on their northward migration in central California. Birds with large volumes of ingested plastic had smaller fat reserves, raising concerns that ingested plastic reduced digestive efficiency or meal size.

Baltz and Morejohn (1976) reported plastic in nine species of seabirds stranded in Monterey Bay, central California, during 1974–1975. All individuals of two species contained plastic: northern fulmar (*Fulmarus glacialis*) and short-tailed shearwater (*Puffinus tenuirostris*). Industrial pellets predominated in these birds, but they were also found to contain pieces of food wrap, foamed polystyrene, synthetic sponge and pieces of rigid plastic. Baltz and Morejohn (1976) speculated that having large volumes of plastic in their stomachs could interfere with the birds' digestion, although they considered that toxic chemicals adsorbed to the plastics posed the greatest threat to bird health. Ohlendorf et al. (1978) showed that plastic ingestion also occurred among Alaskan seabirds.

In the same year that Colton et al. (1974) showed the ubiquitous nature of plastic particles floating in the northwest Atlantic, Wong et al. (1974) reported that plastic pellets were widespread in the North Pacific Ocean. Sampling in 1972, they found that pellets occurred at lower densities (average 300 g km^{-2}) than tar balls, but they outnumbered tar balls northeast of Hawaii, with up to 34,000 pellets km^{-2} (3500 g km^{-2}). Even before this, however, Venrick et al. (1973) had shown that large litter items, at least half of which were made of plastic, were commonly encountered in the North Pacific gyre northeast of Hawaii (roughly 4.2 items km^{-2}) in the area of the now notorious 'North Pacific Garbage Patch'. This is where Moore et al. (2001) recorded densities of more than

300,000 particles km^{-2} in 1999, and where the weight of the plastic was six times that of the associated zooplankton.

Merrell (1980) conducted one of the first detailed studies of beach litter. Working on remote Alaskan beaches, he reported how the amount of plastic litter more than doubled in abundance between 1972 and 1974, increasing from an average density of 122 to 345 kg km^{-1}. Most of this litter came from fisheries operating in the area, but some apparently had drifted more than 1500 km from Asia. At the same time, Jewett (1976) and Feder et al. (1978) found that litter was common on the seabed off Alaska, with plastic items predominating. Merrell (1980) considered that the most obvious impact of beach litter was its aesthetic impact. In terms of biological threats, he speculated that plastic litter might account for the elevated levels of PCBs recorded in rats and intertidal organisms on Amchitka Island, and also suggested that plastics might be a source of phthalates and other toxic compounds into marine systems. Litter also entangled animals, especially seals and seabirds (Merrell 1980), and even terrestrial species were not immune from this problem (Beach et al. 1976).

Merrell (1980) reported the first long-term study of litter accumulation from a 1-km beach on Amchitka Island, Aleutians. He showed that the accumulation rate of litter (average 0.9 kg km^{-1} d^{-1}) varied considerably between sample periods (0.6–2.3 kg km^{-1} d^{-1}), and at a fine temporal scale the amount of litter stranded was a function of recent weather conditions. He also estimated the annual turnover rate of plastic items on the beach by marking gillnet floats, the most abundant litter item on the island, in two successive years. During the intervening year, 41 % of marked floats disappeared (25 % at one beach and 70 % at another beach), but this loss was more than compensated for by new arrivals, with a net increase of 130 %. Merrell (1980) discussed the various factors causing the loss of plastic items from beaches (burial, export inland or out to sea, etc.), and noted the bias introduced by selective beachcombing. Even on remote Amchitka Island, the small Atomic Energy Commission workforce removed certain types of fishing floats within a few days of the floats washing ashore.

The large amounts of litter found in Alaska, coupled with ingestion by seabirds (Ohlendorf et al. 1978) and entanglement of seals (Fowler 1985, 1987), stimulated the first post-graduate thesis on the marine litter problem. Bob Day (1980) studied the amounts of plastic ingested by Alaskan seabirds, in the first community-level study of plastic ingestion. Of the almost 2000 birds from 37 species collected off Alaska from 1969 to 1977, plastic was found in 40 % of species and 23 % of individuals. His main findings were presented in a review paper at the first marine debris conference in 1984 that summarized what was known about plastic ingestion by birds (Day et al. 1985). By that stage, it was clear that the incidence of plastic ingestion varied greatly among taxa, with high rates typically recorded among petrels and shearwaters (Procellariidae), phalaropes (*Phalaropus*) and some auks (Alcidae). Unsurprisingly, generalist foragers that fed near the water surface tended to have the highest plastic loads, although some pursuit-diving shearwaters and auks also contained large amounts of plastic. Plastic items were only found in the stomachs of birds; no visible items passed into the intestines. There was some evidence that at least some species retained plastic particles in their stomachs for considerable periods (up to 15 months), where they slowly eroded. Almost all particles floated in seawater, and comparison of the colors of ingested plastics with observations of the colors of litter items at sea demonstrated that all species favoured more conspicuous items, suggesting they were consumed deliberately. Industrial pellets comprised the majority of plastic items in most species sampled, possibly due to their similarity to fish eggs.

Day et al. (1985) also showed that the incidence of plastic ingestion generally increased over the study period, but patterns were affected by seasonal and age-related differences in plastic loads. Sex had no effect on plastic loads, but immature birds contained more plastic than adults in two of three species where this could be tested. There were also regional differences in plastic loads, with birds from the Aleutian Islands containing more plastic than birds from the Gulf of Alaska, and even lower loads in birds from the Bering and Chukchi Seas. Surveys in the North Atlantic confirmed regional differences in plastic loads in northern fulmars (Bourne 1976; Furness 1985a; van Franeker 1985), paving the way for the use of this species to monitor the abundance and distribution of plastic litter at sea (Ryan et al. 2009; van Franeker et al. 2011; Kühn and van Franeker 2012).

Like Connors and Smith (1982), Day (1980) found weak negative correlations between the amount of ingested plastic and body mass or fat reserves in some species, suggesting a sub-lethal effect on birds. And among parakeet auklets (*Cyclorrhynchus psittacula*), non-breeding adults contained twice as much plastic as breeding adults. However, Day (1980) was quick to point out that the differences in plastic loads could be a consequence of poor body condition or breeding status rather than vice versa. Harper and Fowler (1987) assumed that the negative correlation between the amount of ingested plastic and body mass of juvenile Salvin's prions (*Pachyptila salvini*) stranded in New Zealand in 1966 resulted from starving birds resorting to eat inedible objects such as pumice and plastic pellets. Spear et al. (1995) reported that among a large series of birds collected in the tropical Pacific, heavier birds were more likely to contain plastic, and attributed this to the fact that they fed in productive frontal areas where plastic tends to accumulate (cf. Bourne and Clarke 1984). Among birds that contained plastic, there was a negative correlation between the amount of plastic and body weight, which they interpreted as providing the first solid evidence of a negative relationship between plastic ingestion and body condition (Spear et al. 1995). However, caution must be exercised in such comparisons, given the effects of age and breeding status on the amounts of plastic in seabirds such as petrels that regurgitate accumulated plastic to their chicks (Ryan 1988a).

1.5 Into the Southern Hemisphere

Despite the fact that the first record of plastic ingestion came from the Southern Hemisphere in 1960 (Harper and Fowler 1987), reports of the occurrence of plastics at sea in the Southern Hemisphere generally lagged somewhat behind that in the north. Notable exceptions were the reports of plastic ingestion by turtles in South Africa, where plastic pellets were found in juvenile loggerhead turtles (*Caretta caretta*) in 1968 (Hughes 1970) and a large sheet of plastic was found blocking the intestine of a leatherback turtle (*Dermochelys coriacea*) that died in 1970 (Hughes 1974). The paucity of records of plastic litter from the Southern Hemisphere did not mean that the problem was not as severe in the less industrialized south. Gregory (1977, 1978) reported plastic pellets from virtually all New Zealand beaches, with densities at some beaches estimated at >100,000 pellets m^{-1}, which probably are the highest estimates of industrial pellet densities from any beach. Quite why such high densities were found in a country with a relatively small manufacturing base is unclear. Plastic pellets were also recorded in oceanic waters of the South Atlantic off the Cape in 1979, an area far removed from major shipping lanes and with little industrial activity in adjacent coastal regions (Morris 1980). There was a suggestion that pellets were more abundant west of 12°E (1500–3600 km^{-2}) than closer to the Cape coast (0–2000 km^{-2}), possibly linked to their aggregation in the South Atlantic gyre (cf. Lebreton et al. 2012; Maximenko et al. 2012; Ryan 2014). However, the average density of pellets and other plastic fragments close to the Cape coast was more than 3600 particles km^{-2} (Ryan 1988b), similar to densities reported in oceanic waters of the North Atlantic (Carpenter and Smith 1972; Colton et al. 1974) and North Pacific (Wong et al. 1974). By comparison, the density of pellets and other plastic litter in sub-Antarctic waters south of New Zealand was very low (<100 items km^{-2}, Gregory et al. 1984).

In addition to plastic pellet ingestion by New Zealand prions since the 1960s (Harper and Fowler 1987), rubber bands were found in Antarctic fulmars (*Fulmarus glacialoides*) stranded on New Zealand beaches in 1975 (Crockett and Reed 1976), and during an irruption of Southern Ocean petrels to New Zealand in 1981 all blue petrels (*Halobaena caerulea*) but very few Kerguelen petrels (*Lugensa brevirostris*) contained plastic (Reed 1981). Subsequent studies confirmed the high levels of plastic in blue petrels, despite the species rarely foraging north of the Subtropical Convergence (Ryan 1987a). Sampling in 1981 also showed that at least three petrel species collected in the South Atlantic Ocean contained plastics (Bourne and Imber 1982; Furness 1983; Randall et al. 1983). The incidence was greatest in great shearwaters (*Puffinus gravis*), with 90 % of individuals of this trans-equatorial migrant containing plastic particles, sometimes in large volumes (up to 78 pellets and fragments; Furness 1983). Further surveys even found plastics in Antarctic seabirds, but they were scarce in species that remained south of the Antarctic Polar Front year round compared to migrants that ventured farther north in the non-breeding season (Ryan 1987a; van Franeker and Bell 1988). Beach litter surveys confirmed the presence of plastic wastes in the far south, although the amounts of litter decreased from south temperate to sub-Antarctic and Antarctic locations (Gregory et al. 1984; Gregory 1987; Ryan 1987b).

Bob Furness (1985b) reported the first systematic survey of plastic ingestion by Southern Hemisphere birds for the seabirds of Gough Island, central South Atlantic Ocean. Of the 15 species sampled, 10 contained plastic, and two species had plastic in more than 80 % of individuals sampled. Petrels were again the most affected species, and Furness (1985b) was able to show that this was linked to the structure of their stomachs. The angled constriction between the fore-stomach and gizzard apparently prevents petrels regurgitating indigestible prey remains (except when feeding their chicks). Once again body mass was inversely correlated with the amount of ingested plastic in some species, but Furness (1985b) highlighted the need for controlled experiments to demonstrate an adverse impact of plastic ingestion. Building on this study, Ryan (1987a) showed that 40 of 60 Southern Hemisphere seabird species ingested plastic. Controlling for age and breeding status there was no correlation between plastic load and body condition (Ryan 1987c), but there was a correlation with PCB concentrations (Ryan et al. 1988), and chicks experimentally fed plastic grew more slowly than control birds, because they ate smaller meals (Ryan 1988c). A subsequent experiment showed that marine turtle hatchlings did not increase their food intake sufficiently to offset dietary dilution by an inert substance used to mimic the presence of plastic in their diet (McCauley and Bjorndal 1999).

Although most plastic apparently was ingested directly by the marine vertebrates studied, there was some evidence of secondary ingestion. Eriksson and Burton (2003) collected plastic particles from fur seal scat on Macquarie Island and speculated that they were ingested by lantern fish (*Electrona subaspera*), which were then eaten by the seals. And ingestion was not the only issue reported from the Southern Hemisphere. During the 1970s the rates of entanglement of Cape fur seals (*Arctocephalus pusillus*) in southern Africa (Shaughnessy 1980) were similar to those of northern fur seals in Alaska. The first entangled New Zealand fur seal (*Arctocephalus forsteri*) was observed in 1975 (Cawthorn 1985), and by the late 1970s entanglements of fur seals were recorded as far south as South Georgia (Bonner and McCann 1982). The first entanglements of cetaceans and sharks also were recorded from New Zealand in the 1970s (Cawthorn 1985).

1.6 Aloha—The Marine Debris Conferences

The growing awareness of the accumulation of plastic wastes in marine systems, and their impacts on marine biota, resulted in the Marine Mammal Commission approaching the US National Marine Fisheries Service in 1982 to arrange a workshop on the issue. Given the severity of the problem in the North Pacific Ocean, the task devolved to the Southwest Fisheries Center's Honolulu Laboratory. The Workshop on the Fate and Impact of Marine Debris took place in late November 1984 and was attended by 125 people from eight countries (91 % from the USA, 4 % from Asia, 3 % from Europe and 1 % each from Canada and New Zealand). Given the geographic bias of delegates, most of the 31 papers dealt with the North Pacific, but there were more general papers on the distribution and dynamics of floating litter as well as reviews of entanglement (Wallace 1985), and ingestion by seabirds (Day et al. 1985). The 580-page proceedings, edited by Richard Shomura and Howard Yoshida, appeared laudably fast as a NOAA Technical Memorandum in July 1985. Papers presented at the workshop were divided into three themes: the origins and amounts of marine debris (12 papers), impacts on marine resources (13 papers), and its fate (4 papers). The proceedings concluded with summary documents from working groups addressing each of the three main themes. The workshop emphasized the need to raise awareness of the threat posed by marine litter, and recommended three mitigation initiatives: to regulate the disposal of high-risk plastic items, to promote recycling of fishing nets, and to investigate the use of biodegradable material in fishing gear.

The success of the first marine debris workshop led to plans for a Second International Conference on Marine Debris. However, before this could occur the Sixth International Ocean Disposal Symposium took place in Pacific Grove, California, in April 1986. This was the first symposium in this series to address the dumping of persistent plastic wastes (Wolfe 1987). It was attended by 160 delegates from 10 countries and resulted in a special issue of *Marine Pollution Bulletin* (1987, volume 18, issue 6B). The focus was largely on ship-based sources of marine debris and their impacts, but also addressed incidental bycatch in fishing gear as well as land-based sources of debris. A few papers were repeated from the 1984 Honolulu workshop, and apart from Pruter's (1987) review of litter sources and amounts and Laist's (1987) review of the biological impacts of marine plastics, two of the most important papers dealt with legal approaches and strategies to reduce the amount of plastic entering the sea (Bean 1987; Lentz 1987).

The Second International Conference on Marine Debris was again held in Honolulu in April 1989, attracting over 170 delegates from 10 countries (USA 83 %, Japan 6 %, Canada and New Zealand 3 % each, UK 2 %; all other countries <1 %). It had a more ambitious scope than the first conference, with seven themed sessions following a series of regional overview papers. Whereas the focus of the first meeting was largely on the amounts and impacts of debris, the second conference concentrated more on tackling the problem, with sessions on solutions through technology, law and policy, and education, as well as the first estimates of the economic costs of marine litter. The two-volume, 1274-page proceedings, edited by Richard Shomura and Mary Lynne Godfrey, was again published as a NOAA Technical Memorandum in December 1990 and contained 76 papers plus eight working group reports. The proceedings made numerous recommendations, including nine priority recommendations. Both the first and second conference proceedings are available as internet downloads.

The first two Marine Debris Conferences played a major role in collating information on the marine debris issue. The large numbers of papers in the two proceedings resulted in a spike in publications on the subject (Fig. 1.2). Three further conferences have taken place. The Third International Conference on Marine Debris was held in Miami in May 1994 and had a more Caribbean flavor. It also differed from the two earlier conferences in having only selected papers published from the meeting in a book that aimed to provide a definitive treatment of the marine debris problem (Coe and Rogers 1997). The theme of the conference was 'Seeking Global Solutions', and two-thirds of the papers were devoted to mitigation, with four chapters on the socioeconomics of marine litter, eight chapters addressing at-sea sources, and ten chapters on land-based sources. This reflected the increasing appreciation that not only were diffuse, land-based inputs the major source of marine litter, but that in many ways they were harder to tackle than ship-based sources.

The two most recent Marine Debris Conferences were again held in Honolulu. The fourth conference (August 2000), which focused on the problems posed by derelict fishing gear, attracted 235 people from more than 20 countries, all but one in the Pacific region. The fifth meeting (March 2011) was the largest yet, with more than 450 delegates from across the world, reflecting the mounting concern among civil society regarding the threats posed by marine litter. Entitled '*Waves of Change: Global Lessons to Inspire Local Action*', the conference concluded that despite the challenges inherent in tackling marine debris, the problem is preventable. The summary proceedings, released on the internet after the meeting, included reports from the three working groups established to address the prevention, reduction and management of land-based sources, of at-sea sources, and the removal and processing of accumulated marine debris. The reports highlighted progress made in each of these areas over the last decade, identified remaining challenges, and made recommendations for future action. The conference concluded with the Honolulu Commitment, which called on governmental and non-governmental organisations, industry and other stakeholders to commit to 12 action points, including formulating the Honolulu Strategy to prevent, reduce and manage marine debris. This framework document, sponsored by United Nations Environment Programme and the US National Oceanic and Atmospheric Administration, was released in 2012.

1.7 Mitigation Measures and Long-Term Changes in Marine Litter

One of the major challenges in addressing the marine plastics problem is the diverse nature of plastic products, and the many routes they can follow to enter marine systems (Pruter 1987; Ryan et al. 2009). As a result, a diversity of mitigation measures is needed to tackle the problem. Initial efforts focused on two specific user groups, shipping/fisheries and the plastics industry, at least in part because they are relatively discrete user groups, and thus are more easily addressed (at least in theory). Shipping was a major source of marine litter (Scott 1972; Horsman 1982). Dumping persistent plastic wastes from land-based sources at sea was banned under the Convention on the Prevention of Pollution by Dumping of Wastes and Other Matter (London Dumping Convention, promulgated in 1972; Lentz 1987), but operational wastes generated by vessels were exempt until Annex V of the International Convention for the Prevention of Pollution from Ships (MARPOL, promulagated in 1973) came into force at the end of 1988 (www.imo.org). Since then considerable effort has been expended to ensure there are adequate port facilities to receive wastes from ships (Coe and Rogers 1997). Current signatories to MARPOL Annex V are responsible for more than 97 % of the world's shipping tonnage, but compliance and enforcement remain significant problems (Carpenter and MacGill 2005).

Industrial pellets were another target for early mitigation measures because they were abundant in the environment, often ingested by marine birds and turtles, and only handled by a relatively small group of manufacturers and converters. As early as the 1970s it was clear that improving controls in manufacturing plants could significantly reduce the numbers of pellets entering coastal waters (Kartar et al. 1976). The loss of pellets in wastewater should fall under national water quality control measures, but in most countries the issue has been ignored in favour of chemical pollutants (Bean 1987). As a result, it was left to the plastics industry to initiate efforts to reduce losses of industrial pellets such as Operation Clean Sweep, established in the USA in 1992, and adopted in various guises by many other plastics industry organisations around the world (Redford et al. 1997).

How effective were these measures in reducing litter entering the sea? Although there were some exceptions (e.g. Merrell 1984), amounts of plastic litter at sea increased up to the 1990s, and then appeared to stabilize, whereas quantities on beaches and on the seabed have continued to increase (Barnes et al. 2009; Law et al. 2010). This could result from a decrease in the amounts of litter entering the sea (Barnes et al. 2009), but interpretation is complicated by the difficulty of monitoring marine litter loads, and our rather poor understanding of the rates of degradation and transport between habitats and regions (Ryan et al. 2009). Part of the problem is that mitigation measures may be effective in reducing the proportion of the waste stream reaching the sea, but this decrease may be insufficient to decrease the absolute amount of litter entering the sea, given the ongoing increase in plastic production (Fig. 1.1).

Interaction rates with marine biota provide one way to track the impacts of marine litter, and several studies have focused on the effects of specific mitigation initiatives. For example, the rate of entanglement in Antarctic fur seals (*Arctocephalus gazella*) at South Georgia decreased over the last two decades following active steps to prevent dumping of persistent wastes by vessels operating in the waters around the island. However, some of the decrease can be attributed to changes in seal numbers (Arnould and Croxall 1995; Waluda and Staniland 2013). A similar conclusion was reached by Boren et al. (2006) for New Zealand fur seals, where the decrease in the entanglement rate after 1997 was more likely a result of increasing seal numbers than a decrease in the amounts of litter at sea. Henderson (2001) showed no change in entanglement rates of Hawaiian monk seals (*Monachus schauinslandi*) before and after the implementation of MARPOL Annex V, nor was there a decrease in the rate at which netting washed ashore at the northwest Hawaiian Islands. Page et al. (2004) also showed no change in seal entanglement rates in southeast Australia despite efforts by government and fishing organisations to reduce the amount of litter discarded at sea. However, beach surveys in this region suggested that the implementation of MARPOL Annex V reduced the amounts of litter washed ashore (Edyvane et al. 2004). Ribic et al. (2010)

showed how carefully designed beach litter surveys can detect regional differences in long-term trends in the amounts of stranded litter, with consistent trends in land- and ship-based sources of litter.

Long-term studies of plastic ingestion by seabirds also indicate limited success in tackling the marine litter problem. The rapid increase in the amount of ingested plastic through the 1960s and 1970s (Harper and Fowler 1987; Moser and Lee 1992) stabilized during the 1980s and 1990s (Vlietstra and Parga 2002; Ryan 2008; Bond et al. 2013), but only studies of North Atlantic fulmars show a recent decrease in the amount of ingested plastic (van Franeker et al. 2011). Although the total amount of ingested plastic has tended to remain fairly constant over the last few decades, there has been a marked change in the composition of ingested plastic from pellets to plastic fragments (Vlietstra and Parga 2002; Ryan 2008; van Franeker et al. 2011), suggesting that efforts to reduce the numbers of pellets entering the sea have been at least partly successful. These results mirror the findings of net-samples of plastic litter at sea, which have seen a major increase in the proportion of user fragments and a corresponding decrease in industrial pellets relative to surveys conducted in the 1970s and 1980s (Moore et al. 2001; Law et al. 2010).

1.8 Plastic Degradation and the Microplastic Boom

Although many plastics are remarkably persistent, they are not immune to degradation. Indeed the plastics industry goes to considerable effort to slow the rate of degradation in many applications (Andrady et al. 2003). Ultraviolet (UV) radiation plays a key role in plastic degradation, and because UV light is absorbed rapidly by water, plastics generally take much longer to degrade at sea than on land (Andrady 2003). However, the rate of degradation depends on the ambient temperature as well as polymer type, additives and fillers (Andrady et al. 2003). Carpenter and Smith (1972) observed some degradation in polyethylene pellets collected at sea, but Gregory (1987) inferred degradation occurred more rapidly in stranded plastics, where they were exposed to high levels of UV radiation. The proportion of degraded pellets increased higher up the beach, away from the most recent strandline (Gregory 1987). Little is known about the fate of plastic that sinks to the seafloor; it is widely assumed that plastic is largely impervious to degradation once shielded from UV radiation (Goldberg 1997). However, there is some evidence that plastic fragments may be susceptible to bacterial decay at sea (Harshvardhan and Jha 2013; Zettler et al. 2013).

At the same time that plastics were being recognized as a significant marine pollutant, it was recognized that plastic litter was broken down by photodegradation and oxidation (Scott 1972; Cundell 1974). Scott (1972) reported how some beach litter items became embrittled and were reduced to small particles by very slight pressure. The apparent lack of disintegrated plastic around such items led him to conclude that the particles “had clearly been absorbed rapidly by the environment”

(Scott 1972, p. 36). Gregory (1983) also assumed that this process led to "complete degradation of the plastic pellets and dispersal as dust" (p. 82). However, it was a case of out of sight, out of mind. Thompson et al. (2004) showed that microscopic plastic fragments and fibres are ubiquitous marine pollutants. Together with the high media profile given the Pacific 'garbage patch' (Moore et al. 2001) and similar litter aggregations in other mid-ocean gyres (e.g. Law et al. 2010; Eriksen et al. 2013a), the research by Thompson et al. (2004) was largely responsible for the recent resurgence in interest in the marine litter problem (Fig. 1.2). Like larger plastic items, 'microplastics' (Ryan and Moloney 1990) are now found throughout the world's oceans, including in deep-sea sediments (van Cauwenberghe et al. 2013).

There is ongoing debate as to the size limit for 'microplastics' (Thompson 2015). Some authors take a broad view, including items <5 mm diameter (Arthur et al. 2009), whereas others restrict the term to items <2 mm, <1 mm or even <500 μm (Cole et al. 2011). Andrady (2011) argued the need for three terms: mesoplastics (500 μm–5 mm), microplastics (50–500 μm) and nanoplastics (<50 μm), each with their own set of physical characteristics and biological impacts. Depending on the upper size limit, industrial pellets may or may not be included in the term. But even if we adopt a narrow view, not all microplastics derive from degradation of larger plastic items. Some cosmetics, hand cleaners and air blast cleaning media contain small (<500 μm) plastic beads manufactured specifically for this purpose (Zitko and Hanlon 1991; Gregory 1996), the so-called primary microplastics (Cole et al. 2011). The proportion of primary microplastics in the environment probably is small compared to secondary microplastics, except for some areas of the Great Lakes in the United States (Eriksen et al. 2013b), but it is a largely avoidable source of pollution. Public pressure has already forced one major chemical company to commit to phasing out the use of plastic scrubbers in their products by 2015.

Much of the concern around microplastics concerns their role in introducing persistent organic pollutants (POPs) into marine foodwebs (Cole et al. 2011; Ivar do Sul and Costa 2014). Some of the additives used to modify the properties of plastics are biologically active, potentially affecting development and reproduction (Oehlmann et al. 2009; Meeker et al. 2009). Also, hydrophobic POPs in seawater are adsorbed onto plastic items (Carpenter et al. 1972; Mato et al. 2001; Teuten et al. 2009), and the smaller the particle, the more efficiently they accumulate toxins (Andrady 2011). Thompson et al. (2004) showed that invertebrates from three feeding guilds (detritivores, deposit feeders and filter feeders) all consumed microscopic plastic particles, reinforcing the results of early selectivity experiments demonstrating that filter feeders can consume small plastic particles (De Mott 1988; Bern 1990). Small particles also are eaten by myctophid fish (Boerger et al. 2010), which are an important trophic link in many oceanic ecosystems (Davison and Asch 2011). The subject of POP transfer is explored in more detail by Rochman (2015), but it is worth noting that strict controls on the use of several POPs (e.g. PCBs, HCHs, DDT and its derivatives) have decreased their concentrations on plastic pellets over the last few decades (Ryan et al. 2012). There remain concerns about the health impacts of other compounds whose use is not as strictly regulated (e.g. PBDE, BPA, phthalates, nonylphenol, etc.; Meeker et al. 2009; Oehlmann et al. 2009; Gassel et al. 2013), and even the ingestion of uncontaminated microplastic particles can induce stress responses in fish (Rochman et al. 2013b).

1.9 Summary and Conclusions

Awareness of the threats posed by waste plastics to marine ecosystems developed gradually through the 1960s and 1970s. Most of the environmental impacts of plastic litter were identified in the 1970s and 1980s, resulting in numerous policy discussions and recommendations to decrease the amount of waste plastic entering the environment (Chen 2015). Tightened controls by plastic manufacturers and converters reduced losses of industrial pellets and legislation such as MARPOL Annex V reduced disposal of plastic wastes at sea (although compliance remains problematic in at least some sectors). However, it also became apparent that most litter entering the sea did so from diffuse, land-based sources that are more difficult to control. The rapid increase in global plastic production has resulted in an increase in the amount of plastic items and fragments in marine systems, which in many cases has offset the gains made by reducing losses of industrial pellets and dumping of ship-generated wastes. Plastic is becoming so abundant in some marine systems that it is actually altering the physical properties of the environment (e.g. Carson et al. 2011).

There was a lull in research activity in the 1990s, but the confirmation that microplastics were a ubiquitous marine pollutant in the early 2000s, coupled with publicity around the formation of mid-ocean garbage patches, has stimulated renewed research interest and increased public awareness of the marine litter problem. One of the most urgent current challenges is the need to develop techniques to trace the smallest plastic particles through marine ecosystems, including uptake and release from marine organisms. We also need an improved understanding of the dynamics of waste plastics if we are to monitor the efficacy of mitigation measures (Ryan et al. 2009). Just as we can't interpret the significance of plastic loads in organisms without assessing their turnover rates (Ryan 1988a), we need estimates of transport rates between environments and their biota, and of plastic degradation rates under different environmental conditions. However, we already know enough to say with certainty that the release of waste plastics into the environment is already impacting adversely on marine systems, and affecting human quality of life. Given that plastic litter is, at least theoretically, a wholly avoidable problem, increased effort is needed to stop the inappropriate disposal of waste plastics through a combination of education, product design, incentives, legislation and enforcement.

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